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# Future climate and land-use intensification modify arthropod community structure

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## Abstract

Climate change and land-use intensification pose increasing threats to biodiversity, with climate change expected to eventually surpass other global environmental change drivers and become the greatest threat to biodiversity in the future. Understanding the combined ecological impacts of multiple global change drivers is crucial to predict future scenarios of biodiversity change. However, experimental evidence for the impacts of land-use intensification under current and future climate scenarios is lacking, even though this is imperative for understanding future trajectories of biodiversity in agricultural landscapes. We experimentally tested for the simultaneous effects of land-use intensification and climate change on arthropod biodiversity in a field-scale grassland experiment known as the Global Change Experimental Facility (GCEF). Specifically, we tested whether future scenarios of climate change are likely to exacerbate impacts of land-use intensification on arthropod diversity and abundance across different trophic levels by sampling aboveground arthropod communities in low and high land-use intensity grasslands under current and future climatic conditions. We found that climate change reduced total abundances of arthropods and increased evenness of the whole community, while only having trophic level-specific effects on detritivore abundance and evenness. Land-use intensification reduced abundance of the whole community, predators and detritivores, but only eroded species richness of the whole community and herbivores, with the magnitude of declines in predator and detritivore abundance depending on the climate scenario. Additionally, both land-use intensification and climate altered species composition of the whole community and within the predator, herbivore, and detritivore trophic levels. We show that climate change and land use intensification cause simultaneous shifts in arthropod abundance, species richness, and species composition across trophic levels. Changes in arthropod communities as a result of

climate change and land-use intensification will likely have profound consequences for ecosystem functioning under future environmental conditions.

Keywords: Agricultural intensification, climate change, drought, insects, invertebrates, land-use change, grassland, warming

## **1. Introduction**

Multiple drivers of environmental change pose increasingly significant threats to biodiversity (Fischer et al., 2018; Sala et al., 2000; Vitousek, 1994). Land-use intensification, in particular, is considered to be one of the largest of these threats (Fischer et al., 2018) due to its widespread and dramatic impacts on biodiversity at local and landscape scales (Ekroos et al., 2020; Gossner et al., 2016; Newbold et al., 2015). In grassland ecosystems, processes such as harvesting, and input of fertilizers play a particularly strong role in reducing arthropod abundance and diversity (Ekroos et al., 2020; Humbert et al., 2010). Grasslands cover approximately 30% of the Earth's land surface and are critical for delivering a wide range of ecosystem services, including forage production for livestock, carbon sequestration, and biodiversity conservation (Bengtsson et al., 2019; Pilgrim et al., 2010). While land-use intensification may currently pose the greatest and most acute environmental threats to grasslands, climate change is expected to eventually surpass other global environmental change drivers and become the greatest threat to biodiversity in the future (Millennium Ecosystem Assessment, 2005).

In this century, mean surface temperatures are expected to increase by at least 1.5 °C to 4.8 °C compared to preindustrial levels as a result of human activities, depending on greenhouse gas concentrations in the atmosphere (IPCC, 2014). Shifts in global precipitation regimes are also expected which, in combination with warming, will lead to more frequent drought events and continue to intensify threats to global biodiversity (Tilman and El Haddi, 1992). These global change drivers, however, are unlikely to have independent effects on ecosystems. Instead, it is crucial to consider interactions between global change effects, such as climate change and land-use, to understand the synergistic or antagonistic effects these drivers may have on ecological communities (Binzer et al., 2016; Oliver and Morecroft, 2014). Indeed, the combination of changing climate with land-use intensification could potentially explain the suspected dramatic decline in insect abundance over the last decades, but experimental evidence is crucial to identify the drivers of these observed changes in arthropod densities (Dormann et al., 2008; Hallmann et al., 2017; Soroye et al., 2020).

Abundance, diversity, and composition are important characteristics that collectively describe different aspects of community structure and are all strongly linked to ecosystem functioning (Barnes et al., 2018; Hooper et al., 2005; Soliveres et al., 2016). These components can therefore be used as indicators of ecosystem responses to global environmental changes. Changes in diversity, namely whether certain species are entirely extirpated from a system or the relative dominance of species changes in response to environmental stress, can be determined by quantifying species richness and evenness, respectively. Diverse and abundant arthropod communities provide a variety of ecosystem functions that contribute to the supply of ecosystem services and are therefore highly beneficial for humans (Cardinale et al., 2012; Chapin III et al., 2000). These include processes regulating food security or soil fertility through pollination, natural pest control, and decomposition (Altieri, 1999; Barnes et al., 2020; Doran and Zeiss, 2000; Kremen et al.,

2007; von Berg et al., 2009), which may be under threat as arthropod communities respond to multiple environmental change drivers.

Recent studies have documented alarming declines of arthropods over the past decades (Hallmann et al., 2017; Leather, 2018; Seibold et al., 2019; van Klink et al., 2020), though there is still considerable debate over the validity and generality of these worrying trends (e.g. Crossley et al., 2020). There is considerable evidence from space-for-time studies and time-series studies that arthropod declines are linked to land-use intensification, (including shifts in habitat structure, plant-diversity and fertilizer use) and also some evidence for negative impacts of climate change (Dormann et al., 2008; Ekroos et al., 2020; Gossner et al., 2016; Hendrickx et al., 2007; Seibold et al., 2019). Additionally, organisms from different trophic levels may respond differently to global change drivers (Dormann et al., 2008) and thus the processes linked to these groups, such as decomposition or pest-control, may follow similar patterns. However, experimental tests of combined land-use intensification and climate change impacts on arthropod communities are lacking. Due to the highly complex nature of the direct and interactive impacts of these combined environmental change drivers, it is inherently difficult to delineate clear hypotheses of how multitrophic species assemblages will respond when they combine. Therefore, experimental studies that can isolate and identify the effects of these drivers and their interactions are imperative to begin to understand the future consequences of these combined global change drivers for arthropod biodiversity.

To determine whether future scenarios of climate change are likely to exacerbate impacts of land-use intensification on arthropod communities (Mantyka-Pringle et al., 2015; Oliver and Morecroft, 2014) spanning multiple trophic levels, we test for responses in abundance, species richness, species evenness and community composition to land-use intensification under current and future climatic conditions. To do so, we use an agricultural

field-scale global change experimental facility in Germany (the GCEF; Schädler et al., 2019), to investigate the combined effects of climate change and land-use intensification on grasslands by simulating a future climate change scenario under field conditions at different levels of land-use intensification. We test for interactive effects of climate change and land-use intensification, whereby we expected land use impacts on invertebrate communities to be exacerbated under the future climate scenario. We hypothesised that arthropod abundance, species richness, and evenness would decline in intensively managed grasslands and under the future climate scenario. In particular, we expected that the simulated global change drivers would affect species richness, evenness and species composition due to differential, trait-dependent responses of species (Simons et al., 2015). In that vein, we also expected that the strength of these responses would vary among trophic levels due to their varying sensitivities to environmental change drivers.

## **2. Materials and Methods**

### **2.1 Study Site**

We used the experimental set-up of the Global Change Experimental Facility (GCEF) in Bad Lauchstädt, Saxony-Anhalt, Germany. The area is characterized by sub-continental climate, with a mean annual temperature of 9.7°C (1993–2013) and mean annual precipitation of 525 mm (1993–2013) (Schädler et al., 2019). The GCEF experiment started in 2014 and consists of ten experimental units (80 × 24 m), where each is covered by a steel frame. Five of the experimental units are influenced by current climatic conditions where the steel frame only serves as a control for climate manipulation. The remaining five units are influenced by a future climate change treatment whereby the walls and roof are added to the steel frames and

are closed overnight. In doing so, this effectively results in a passive temperature increase of the daily mean temperature by 0.55 °C and also manipulates precipitation patterns. In spring and autumn, the quantity of rain is artificially increased by ca. 10 % with irrigation, and the summer precipitation is reduced by ca. 20 %. The climate change treatment is based on the climatic conditions projected for central Germany in 2070-2100, based on different dynamical regional climate models (for details, see Schädler et al., 2019).

Within each of the current (control) and climate change manipulation units, we utilized two different land-use intensity treatments applied on 16 x 24 m plots: i) low intensity managed grasslands and ii) high intensity managed grassland (see Figure 1). While both land-use treatments were managed by mowing to simulate harvesting, the two land-use treatments differed in their initial sown plant species richness, harvest regime, and fertilizer use. The high intensity managed grassland plots had five sown grass species, were harvested four times per year (with the first harvest in April), and were moderately fertilized. In contrast, the low intensity managed grassland had 56 plant species sown initially, were mown two times per year (with the first harvest in June), and had no fertilizers applied (see Schädler et al. (2019) for more details).

## **2.2 Sampling and Species Identification**

In order to sample as many species as possible despite different phenologies across the growing season, arthropods were sampled four times in total, in April, May, early July and late July of 2017. We collected all aboveground arthropods in a 25 cm x 25 cm x 100 cm sampling cage (0.25 m<sup>3</sup> volume) using a suction sampler (EcoVac, ecoTech Umwelt Meßsysteme). The entire cage was exhaustively sampled (i.e. ground vegetation and flying arthropods) for approximately one minute until no more arthropods were visible in the cage,



which has been shown to be highly effective for sampling aboveground grassland arthropods (Brook et al., 2008). Organisms were then stored in a cooling box for transport to the laboratory where they were first stored at -20 °C and then transferred into 70 % ethanol.

### **2.3 Species and trophic group identification**

All arthropods (excluding larval and pupal stages) were first sorted to order level, of which five of the most abundant major taxonomic groups (Araneae, Auchenorrhyncha, Coleoptera, Diptera, and Hymenoptera; see Table S1 in supporting information for a complete list of collected taxa) were further identified to family, genus, or species level when possible by specialist taxonomists (see Acknowledgements). To quantify the effect of climate change and land-use intensity on arthropod abundance and diversity, we quantified abundance as the accumulative sum of all individuals across the 4 temporal samples per cage, observed species richness (i.e., the total number of unique species collected across the 4 temporal samples), expected species richness using the jackknife2-estimator (accounting for possible undersampling of the communities), and species evenness as response variables. Evenness was calculated with the  $E_{var}$  index (Smith and Wilson, 1996), as this evenness index is independent of species richness. In order to quantify the changes in relative abundances and composition of arthropod communities between the climate change and land-use intensity treatments, we assessed the dissimilarity of communities among treatments using the “metaMDS” function from the vegan package using the Bray-Curtis Dissimilarity index.

In addition to the whole community response, we tested the effect of climate change and land-use intensification on different trophic groups. Therefore, all individuals of the five taxonomic groups were ascribed to trophic groups. The selected trophic groups were predators (arthropods that consume or parasitize animals), herbivores (arthropods that

consume or parasitize plant tissues or fluids), omnivores (arthropods that feed on different trophic levels, such as animals, plant material or dead organic matter) and detritivores (arthropods that consume dead organic matter). Literature was searched for ascribing trophic groups to species, genus, family or orders of arthropods, with the highest possible taxonomic resolution (see Table S2 for more information). For analyses at both the whole-community and trophic-group-level, only collected specimens that were identified to species were included in analyses of species richness, evenness, and composition, whereas all specimens from the five most abundant taxonomic groups were included in analyses of arthropod abundances.

## **2.4 Statistical Analysis**

We used generalized linear mixed effects models (GLMM) with climate change, land-use intensification, and their interaction specified as fixed effects, along with experimental block as a random effect (a random-intercept model, using the ‘lme4’ package in R). We modelled the responses of arthropod abundance and species richness to the global change treatments on a Poisson error distribution, except where overdispersion was detected in which case we used a negative binomial error distribution. Species evenness was modelled on a Gaussian error distribution and was log-transformed in order to meet the assumptions of normality and ensure homoscedasticity of variance. We applied post hoc comparisons of estimated marginal means between climate treatments and between land-use intensity treatments nested within climate treatments (using the ‘emmeans’ package in R) in order to graphically highlight significant effects detected in the GLMMs.

To determine the effects of climate change and land-use intensification on dissimilarity in species composition, we applied a permutational multivariate analysis of

variance (PERMANOVA) using the ‘adonis’ function in the ‘vegan’ package, where we tested the effects of climate change and land-use intensification on the dissimilarity of arthropod communities based on Bray-Curtis dissimilarity. For each model, we specified ‘strata’ as experimental block, to account for variation among blocks in the experimental design. Finally, where statistically significant effects of experimental treatments were detected from the PERMANOVA, we conducted similarity percentages (SIMPER) analysis using the ‘simper’ function to identify the set of species that contributed most to whole-community dissimilarity between the land use and climate treatments. All statistical analyses were performed using R version 4.0.2 (R Core Team, 2020).

### **3. Results**

We collected a total of 12,899 arthropods, of which 289 species were identified from the five investigated taxonomic groups: Araneae (spiders; 1,037 individuals), Auchenorrhyncha (leafhoppers; 2,640 individuals), Coleoptera (beetles; 2,137 individuals), Diptera (flies; 1,429 individuals), and Hymenoptera (2,640 individuals). The highest species number was found for beetles with 141 species, followed by flies (84 species), spiders (30 species), leafhoppers (24 species), and Hymenoptera with 10 species and 95 families (some individuals from Araneae, Auchenorrhyncha, Coleoptera, and Diptera could only be identified to the genus level, and the majority of Hymenoptera were identified only to family level). Of the whole community, the 9,731 individuals that could be identified to species were categorized into trophic groups. Herbivores were the most abundant group, with 4,551 individuals and 166 species. Predators were the second most abundant group with 2,750 individuals and 76 species, followed by omnivores with 1,705 individuals and 24 species. Detritivores were the

least abundant group in our sampling with 725 individuals and 23 species (see Table S2 for more details).

### **3.1 Whole-community responses to climate change and land-use intensification**

We found varying effects of climate change and land-use intensification on abundance, species richness, and evenness of arthropod communities in the Global Change Experimental Facility. The abundance of the whole arthropod community was negatively affected by both climate change and land-use intensification. We found that total arthropod abundance was, on average, 20.1 % lower under future climatic conditions when compared to current climatic conditions. We also found an even stronger decline in community abundance in response to land-use intensification, with 37.5% lower average arthropod abundance in high versus low-intensity grassland plots. (Table 1; Figure 2A). In contrast to abundance, arthropod species richness was significantly affected only by land-use intensity, for which we found an average decline in mean species richness of 19.5 % with land-use intensification (Table 1, Figure 2B). In addition to observed species richness, we found that expected species richness followed the same response to land-use intensification (Figure S1). Community evenness was only significantly affected by climate change, where we found an average increase in evenness from 0.48 ( $\pm$  0.04 SD) under current climatic conditions to 0.56 ( $\pm$  0.03 SD) under future climatic conditions (Table 1, Figure 2C). As this result appeared rather counterintuitive, given that we expected certain species to perform better than others under future climatic conditions (thus leading to reduced species evenness), we analyzed rank abundance curves for both climate treatments to determine the cause of this positive response. These analyses revealed that the abundances of dominant species were strongly reduced with climate change

(Figure S2). This resulted in a more even distribution of abundance across species, hence notable dominance by one or a few species was lower in the future climate treatments.

### **3.2 Effects of climate change and land-use intensification on trophic groups**

We tested how climate change and land-use intensity affect abundance, species richness, and evenness of four major trophic groups found in aboveground central European grassland arthropod communities: predators, herbivores, omnivores and detritivores. Consistent with the whole arthropod community, we found that the abundance of predatory arthropods decreased with land-use intensification. From the low to high-intensity land-use treatments, predator community abundance decreased by 45.2 % under current climatic, which was stronger than the 32.5 % decline under future climatic conditions (Table 2, Figure 3A). This diminished response to land use under future climatic conditions was supported by a significant interaction between climate and land-use intensity (Table 2, Figure 3A). We did not detect an effect of climate change and land-use intensification on abundance of herbivores (but a trend toward declining abundance under intensive land use and future climate) and there was similarly no discernible effect of the global change treatments on the community structure of omnivores (Table 2, Figure 3C). In contrast, detritivores were the only group where we found a significant main effect of climate change on abundance, whereby detritivore community abundance decreased on average by 49.6 % from current to future climate conditions. Similar to predators, the effects of land-use intensification on detritivore communities were found to vary under current versus future climatic conditions, such that reductions in detritivore mean abundance were significantly larger under current (55.7 % decline) versus future (34.7 % decline) climatic conditions (Table 2, Figure 3D).

Of all four trophic groups, herbivores had the highest species richness with 165 species in total. This was the only group to show significant changes in species richness in response to the simulated global change, with a reduction in average herbivore species richness of 30.2 % from low to high land-use intensity plots (Table 2, Figure 3F). Furthermore, detritivore evenness increased significantly from the current to future climate scenario (Table 2, Fig. 3L). We also found a marginally significant increase in herbivore species evenness ( $P = 0.052$ ) from current to future climate treatments, but no discernible influence of land-use intensification on herbivore evenness (Table 2, Figure 3J). While detritivore evenness showed a significant increase with land-use intensification and from current to future climate treatments (Table 2, Fig. 3L), predator and omnivore evenness were not affected by either of the global change drivers (Table 2, Figure 3I & K). Of particular note, omnivores appeared to be completely unaffected by both global change drivers, as we found no significant shifts in abundance, species richness, or evenness.

### 3.3 Shifts in community composition

We found that community composition was significantly affected by both climate change and land-use intensification (Table 3, Figure 4A), though effects of land-use intensification were clearly stronger than effects of climate change for the overall arthropod communities. While significant differences in community composition between current and future climate treatments were also found, these differences were negligible in comparison to those found in response to land-use intensity (Table 3, Figure 4A). Hence, communities sampled from the same land-use intensities but different climatic conditions were more similar than communities from different land-use intensities and different climate treatments. When looking at individual species contributions to these compositional changes, the SIMPER

analysis revealed generally low and varied contributions of almost all species to community dissimilarity. The only exception was for two herbivore species, *Javesella pellucida* (Delphacidae) and *Aphrodes makarovi* (Cicadellidae), that had the highest contributions to mean dissimilarity between low versus high land-use intensity (5 % and 4 %, respectively; Table S3) and current versus future climate treatments (5 % and 3 %, respectively; Table S3). Consistent with the community-level shifts in species' relative abundances, the composition of predator, herbivore and detritivore communities were also significantly affected by climate change and land-use intensification (Table 3, Figure 4B, C & D). Whole community and herbivore community composition showed very similar patterns, as they were particularly strongly affected by land-use intensification with especially distinct communities in low versus high-land use intensity plots, but only small differences between climate treatments (Figure 4A & C). Similarly, predator and detritivore communities also varied significantly across land-use intensities and climate treatments, though land use was, again, the most important driver of compositional change (Figure 4B & D). In line with the other metrics describing communities, the only exception were omnivores, where we did not find any significant effects of climate or land use on community composition (Figure 4E).

#### 4. Discussion

By applying interacting treatments of climate change and land-use intensification in a field-scale experimental grassland system, our study provides evidence that both land-use intensification and climate change appear to simultaneously and additively impact arthropod community structure. There was some evidence that the strength of future impacts of land-use intensification on arthropods will be influenced by climate change, whereby future climate reduced the impacts of land-use intensification for predatory and detritivorous arthropod

abundance. Under current and future climatic conditions, total arthropod abundance was significantly lower in high-intensity land use plots, which lends support to the notion that land-use intensification is likely to be one of the major drivers of insect declines (Hallmann et al., 2017; Seibold et al., 2019). Furthermore, we found significant declines in the species richness of whole arthropod communities and herbivores from low to high-intensity land use. Thus, with intensification of land use and under current trajectories of climate change, future grasslands are likely to be characterized by significantly reduced abundance and species richness of arthropod predators, herbivores and detritivores, suggesting that associated ecosystem functions may become threatened as a consequence.

Our hypothesis that future climatic conditions would reduce overall arthropod abundance was mostly supported, with clear declines at the whole community level and for detritivores. In contrast to our findings, past empirical studies have shown increased temperature to lead to higher arthropod abundances (de Sassi et al., 2012; Robinson et al., 2018). However, the observed positive effects of warming in these studies were mainly plant-mediated, where shifts in the plant community as a result of warming increased arthropod abundance. Unlike these past warming experiments, the climate change treatment in our experiment combined a temperature and precipitation manipulation to simulate the multi-faceted nature of climate change (IPCC, 2014). Therefore, it is possible that the drought effect of the climate treatment in our experiment drove the decrease in arthropod abundance, either directly by water stress or indirectly through plant-mediated drought effects that alter the availability and quality of resources for arthropod communities (Huberty and Denno, 2004). However, due to the multi-faceted nature of our climate change treatment, we were not able to disentangle the partial effects of temperature and drought in this study.

The abundance of the whole community and of predators and detritivores decreased with land-use intensification under both current and future climatic conditions. However, the



magnitude of predator and detritivore arthropod declines varied between the climate scenarios, with a stronger decrease in abundance from low to high land-use intensity under current climatic conditions. Here, climate change appeared to limit the maximum abundance of arthropod predators and detritivores in low-intensity land use plots due to drought and higher temperatures, which therefore constrained the potential response of these arthropods to land-use intensification. Hence, arthropod predator and detritivore communities that generally benefit from less intensive management practices show lower abundances under future climate. Altogether, this result suggests that low-intensity grasslands may act as refuges for biodiversity (Öckinger and Smith, 2006), which could be especially threatened under future climate scenarios. Further research on the combined impacts of climate and land-use intensification in non-experimental systems will be needed to determine if this prediction holds true in real grasslands.

Arthropod abundance is typically positively correlated with species richness (Schuldt et al., 2019), so we expected to find similar patterns of species richness in response to the two global change drivers as for abundance. Indeed, species richness of the whole-community and herbivores decreased with land-use intensification, but was not affected by climate change. Specialist primary consumers may be limited by the presence of their preferred resource plants (Koricheva et al., 2000) and, therefore, plant diversity can limit consumer abundance and diversity (Barnes et al., 2020; Borer et al., 2012). Furthermore, habitats with higher plant diversity provide a more heterogeneous habitat structure, which promotes arthropod diversity (Schuldt et al., 2019; Thomas and Marshall, 1999). These mechanisms could explain the observed decline in detritivore abundance and herbivore species richness, as plant diversity was lower in the intensively managed grassland treatments in our study. Increased harvest frequency in the intensively used grasslands may also be a driver of decreasing arthropod abundance and diversity, through direct mechanical habitat destruction and damage to the

arthropods (Humbert et al., 2010; Klaus et al., 2013). These plant-mediated changes in herbivore diversity and detritivore abundance could indirectly affect higher trophic levels (Scherber et al., 2010), which might explain the decrease of predator abundance, in addition to mechanical damage through harvesting. Furthermore, the timing of the first harvest event plays an important role for the arthropod community; harvesting in early spring, which was done in intensively used grasslands, more strongly affects the less mobile stages of insects inhabiting the vegetation, which could explain observed reductions of population size and diversity (van Klink et al., 2019).

As environmental change likely filters for species with certain functional traits (de Sassi et al., 2012), we expected a decrease in evenness with global change, with increasing dominance of species that benefit from the environmental conditions caused by land-use intensification and climate change. Surprisingly, we found that evenness increased with climate change and that land-use intensification had no effect on the evenness of the whole community. The climate change response of evenness is likely caused by the loss of individuals from dominant species' populations, as we found strong declines in dominant species with high abundances under current climatic conditions (indicated by rank abundance curves; Figure S2). These dominant species can play an important role in ecosystem functioning as they tend to contribute strongly to the flow of energy in the community (Hillebrand et al., 2008). Under stable environmental conditions, low evenness within a community could theoretically lead to higher productivity of the system, as species that are well adapted to certain conditions dominate the community (Hooper et al., 2005). However, under changing environmental conditions, such as climate change or land-use intensification, higher evenness can become beneficial due to the higher capacity to adapt to the new conditions (Norberg et al., 2001). Detritivores were the only trophic group that showed a change in species evenness with climate change and with land-use intensification, which,

surprisingly, were positive responses. This was likely due to reductions of more abundant species, combined with no significant losses in species richness in intensive grasslands and future climate treatments, thereby leading to increased evenness despite no significant changes in detritivore species richness.

Overall, the composition of arthropod communities was influenced by both climate change and land-use intensification. Predator, herbivore, and detritivore community composition showed the same patterns as the whole community in response to climate change and land-use intensification, with a consistently stronger influence of land-use intensification. Here, land-use intensification affected both the abundance and presence of species. In particular, different environmental conditions favour different species, which likely drove turnover from species that benefited from low land-use intensity to species that benefited from or could persist in high land-use intensity systems, as well as from those that benefited from current versus future climatic conditions. Furthermore, changes in the relative abundance of species (without species turnover) could have also driven the observed dissimilarity among communities, whereby certain species' abundances responded negatively or positively to changing environmental conditions. Interestingly, we found little evidence that any particular species were responsible for whole-community shifts observed in response to land use and climate change, as individual species contributions were generally minor and varied. This result likely indicates that specific traits (e.g. trophic group) may be more important than species identity for explaining responses in composition of whole arthropod communities to multiple global change drivers. In contrast to herbivores, predators, and detritivores that all exhibited significant shifts in community composition, omnivores showed no responses in community composition, species richness, evenness, or abundance to the experimental global change drivers investigated in this study. The invariance of omnivores in response to the climate and land-use treatments could be due to their ability to switch among

different resources, thereby allowing them to compensate for rarity or loss of certain resource components under these global change scenarios (Coll and Guershon, 2002). However, given the generally low numbers of omnivores across our treatments, any conclusions from these data must be made with caution and further investigation is needed to determine the underlying reasons for these findings.

In real-world ecosystems that will be increasingly influenced by climate change and land-use intensification simultaneously, mechanisms such as eco-evolutionary processes and adaptations, range shifts, and species invasion will strongly impact ecological communities (Scheffers et al., 2016) and may alter them in a manner which we were not able to cover here. Due to the spatially and temporally restricted experimental design of the Global Change Experimental Facility (GCEF), we were unable to account for these important aspects of changing ecosystems. The habitats of the GCEF can only be colonized by arthropods from the surrounding species pool, which is shaped by past and current climate and land use (see Figure 1). However, it allows us to study how ecological communities formed by species adapted to current climatic conditions could potentially perform under different land-use intensities and climate scenarios. It is also important to note that seasonal and inter-annual variation in climatic conditions is expected to increase in the future, which will likely have profound impacts on ecosystems (Thompson et al., 2013). We only partially account for, but do not explicitly model, temporal variability in our study, which could mean that we have missed seasonal and longer-term effects of the simulated global change drivers. Nevertheless, we anticipate that the effects shown in our study may be even stronger under future conditions in real-world (non-experimental) grassland systems, as climate extremes increase in the future and the ecological filtering of local communities will have a feedback effect on the species pool.

We show that under current climatic conditions, land-use intensification strongly impacts arthropod species richness, evenness, abundance, and community composition. In the future, however, grasslands are projected to experience a combination of climate change and further intensification of land use. While climate change, alone, reduces abundances of arthropods and increases evenness, we show that climate change and land-use intensification cause simultaneous shifts in arthropod abundance, species richness, and species composition, that appear to be additive in nature and will likely have profound consequences for ecosystem functioning (Cardinale et al., 2006). Under future climate scenarios, reducing intensive land-use practices will be imperative to maintain the natural supply of ecosystems services provided by arthropod biodiversity in agroecosystems.

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## **Authors' contributions**

ES, BCR and ADB conceived and designed the study, EHS carried out the field and laboratory work, EHS, UB, BR and ADB analyzed the data, and all authors interpreted the results. EHS wrote a first draft and all authors contributed substantially to the writing.

## **Data availability statement**

Upon acceptance, data will be uploaded to an online repository.

## **Literature**

Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. *Agric. Ecosyst. Environ.* 74, 19–31. [https://doi.org/10.1016/S0167-8809\(99\)00028-6](https://doi.org/10.1016/S0167-8809(99)00028-6)

Barnes, A.D., Jochum, M., Lefcheck, J.S., Eisenhauer, N., Scherber, C., O'Connor, M.I., de Ruiter, P., Brose, U., 2018. Energy flux: The link between multitrophic biodiversity and ecosystem functioning. *Trends Ecol. Evol.* 33, 186–197. <https://doi.org/10.1016/j.tree.2017.12.007>

Barnes, A.D., Scherber, C., Brose, U., Borer, E.T., Ebeling, A., Gauzens, B., Giling, D.P., Hines, J., Isbell, F., Ristok, C., Tilman, D., Weisser, W.W., Eisenhauer, N., 2020. Biodiversity enhances the multitrophic control of arthropod herbivory. *Sci. Adv.* 6, 1–9. <https://doi.org/10.1126/SCIADV.ABB6603>

506 Bengtsson, J., Bullock, J.M., Egoh, B., Everson, C., Everson, T., O'Connor, T., O'Farrell,  
 507 P.J., Smith, H.G., Lindborg, R., 2019. Grasslands—more important for ecosystem  
 508 services than you might think. *Ecosphere* 10, e02582.  
 509 <https://doi.org/10.1002/ECS2.2582>

510 Binzer, A., Guill, C., Rall, B.C., Brose, U., 2016. Interactive effects of warming,  
 511 eutrophication and size structure: Impacts on biodiversity and food-web structure. *Glob.*  
 512 *Chang. Biol.* 22, 220–227. <https://doi.org/10.1111/gcb.13086>

513 Borer, E.T., Seabloom, E.W., Tilman, D., 2012. Plant diversity controls arthropod biomass  
 514 and temporal stability. *Ecol. Lett.* 15, 1457–1464. <https://doi.org/10.1111/ele.12006>

515 Brook, A.J., Woodcock, B.A., Sinka, M., Vanbergen, A.J., 2008. Experimental verification of  
 516 suction sampler capture efficiency in grasslands of differing vegetation height and  
 517 structure. *J. Appl. Ecol.* 45, 1357–1363. [https://doi.org/10.1111/j.1365-](https://doi.org/10.1111/j.1365-2664.2008.01530.x)  
 518 [2664.2008.01530.x](https://doi.org/10.1111/j.1365-2664.2008.01530.x)

519 Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani,  
 520 A., Mace, G.M., Tilman, D., A. Wardle, D., Kinzig, A.P., Daily, G.C., Loreau, M.,  
 521 Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and  
 522 its impact on humanity. *Nature* 486, 59–67. <https://doi.org/10.1038/nature11373>

523 Cardinale, B.J., Srivastava, D.S., Duffy, J.E., Wright, J.P., Downing, A.L., Sankaran, M.,  
 524 Jouseau, C., 2006. Effects of biodiversity on the functioning of trophic groups and  
 525 ecosystems. *Nature* 443, 989–992. <https://doi.org/10.1038/nature05202>

526 Chapin III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L.,  
 527 Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C., Diaz, S., 2000.  
 528 Consequences of changing biodiversity. *Nature* 405, 234–242.

529 Coll, M., Guershon, M., 2002. Omnivory in Terrestrial Arthropods: Mixing Plant and Prey  
530 Diets. *Annu. Rev. Entomol.* 47, 267–297.  
531 <https://doi.org/10.1146/annurev.ento.47.091201.145209>

532 Crossley, M.S., Meier, A.R., Baldwin, E.M., Berry, L.L., Crenshaw, L.C., Hartman, G.L.,  
533 Lagos-kutz, D., Nichols, D.H., Patel, K., Varriano, S., Snyder, W.E., Moran, M.D.,  
534 2020. No net insect abundance and diversity declines across US Long Term Ecological  
535 Research sites. *Nat. Ecol. Evol.* 1368–1376. <https://doi.org/10.1038/s41559-020-1269-4>

536 de Sassi, C., Lewis, O.T., Tylianakis, J.M., 2012. Plant-mediated and nonadditive effects of  
537 two global change drivers on an insect herbivore community. *Ecology* 93, 1892–1901.  
538 <https://doi.org/10.1890/11-1839.1>

539 Doran, J.W., Zeiss, M.R., 2000. Soil health and sustainability: managing the biotic  
540 component of soil quality. *Appl. Soil Ecol.* 15, 3–11. [https://doi.org/10.1016/S0929-](https://doi.org/10.1016/S0929-1393(00)00067-6)  
541 [1393\(00\)00067-6](https://doi.org/10.1016/S0929-1393(00)00067-6)

542 Dormann, C.F., Schweiger, O., Arens, P., Augenstein, I., Aviron, S., Bailey, D., Baudry, J.,  
543 Billeter, R., Bugter, R., Bukáček, R., Burel, F., Cerny, M., Cock, R. De, Blust, G. De,  
544 DeFilippi, R., Diekötter, T., Dirksen, J., Durka, W., Edwards, P.J., Frenzel, M.,  
545 Hamersky, R., Hendrickx, F., Herzog, F., Klotz, S., Koolstra, B., Lausch, A., Coeur, D.  
546 Le, Liira, J., Maelfait, J.P., Opdam, P., Roubalova, M., Schermann-Legionnet, A.,  
547 Schermann, N., Schmidt, T., Smulders, M.J.M., Speelmans, M., Simova, P., Verboom,  
548 J., Wingerden, W. van, Zobel, M., 2008. Prediction uncertainty of environmental change  
549 effects on temperate European biodiversity. *Ecol. Lett.* 11, 235–244.  
550 <https://doi.org/10.1111/j.1461-0248.2007.01142.x>

551 Ekroos, J., Kleijn, D., Batáry, P., Albrecht, M., Báldi, A., Blüthgen, N., Knop, E., Kovács-  
552 Hostyánszki, A., Smith, H.G., 2020. High land-use intensity in grasslands constrains



553 wild bee species richness in Europe. *Biol. Conserv.* 241.  
554 <https://doi.org/10.1016/j.biocon.2019.108255>

555 Fischer, M., Rounsevell, M., Torre-Marín Rando, A., Mader, A., Church, A., Elbakidze, M.,  
556 Elias, V., Hahn, T., Harrison, P.A., Hauck, J., Martín- López, B., Ring, I., Sandström,  
557 C., Sousa Pinto, I., Visconti, P., Zimmermann, N.E., Christie, M., 2018. Biodiversity  
558 and Ecosystem Services for Europe and Central Asia. Secretariat of the  
559 Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services,  
560 Bonn, Germany.

561 Gossner, M.M., Lewinsohn, T.M., Kahl, T., Grassein, F., Boch, S., Prati, D., Birkhofer, K.,  
562 Renner, S.C., Sikorski, J., Wubet, T., Arndt, H., Baumgartner, V., Blaser, S., Blüthgen,  
563 N., Börschig, C., Buscot, F., Diekötter, T., Jorge, L.R., Jung, K., Keyel, A.C., Klein, A.-  
564 M., Klemmer, S., Krauss, J., Lange, M., Müller, J., Overmann, J., Pašalić, E., Penone,  
565 C., Perović, D.J., Purschke, O., Schall, P., Socher, S.A., Sonnemann, I., Tschapka, M.,  
566 Tschardt, T., Türke, M., Venter, P.C., Weiner, C.N., Werner, M., Wolters, V., Wurst,  
567 S., Westphal, C., Fischer, M., Weisser, W.W., Allan, E., 2016. Land-use intensification  
568 causes multitrophic homogenization of grassland communities. *Nature* 540, 266–269.  
569 <https://doi.org/10.1038/nature20575>

570 Hallmann, C.A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W.,  
571 Müller, A., Sumser, H., Hörden, T., Goulson, D., de Kroon, H., 2017. More than 75  
572 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS One*  
573 12, e0185809. <https://doi.org/10.1371/journal.pone.0185809>

574 Hendrickx, F., Maelfait, J.P., Wingerden, W.V., Schweiger, O., Speelmans, M., Aviron, S.,  
575 Augenstein, I., Billeter, R., Bailey, D., Bukacek, R., Burel, F., Diekötter, T., Dirksen, J.,  
576 Herzog, F., Liira, J., Roubalova, M., Vandomme, V., Bugter, R., 2007. How landscape

structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes. *J. Appl. Ecol.* 44, 340–351.  
<https://doi.org/10.1111/j.1365-2664.2006.01270.x>

Hillebrand, H., Bennett, D.M., Cadotte, M.W., 2008. Consequences of dominance: a review of evenness effects on local and regional ecosystem processes. *Ecology* 89, 1510–1520.  
<https://doi.org/10.1890/07-1053.1>

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecol. Monogr.* 75, 3–35. <https://doi.org/10.1890/04-0922>

Huberty, A.F., Denno, R.F., 2004. Plant water stress and its consequences for herbivorous insects: a new synthesis. *Ecology* 85, 1383–1398. <https://doi.org/10.1890/03-0352>

Humbert, J.-Y., Ghazoul, J., Sauter, G.J., Walter, T., 2010. Impact of different meadow mowing techniques on field invertebrates. *J. Appl. Entomol.* 134, 592–599.  
<https://doi.org/10.1111/j.1439-0418.2009.01503.x>

IPCC, 2014. IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. The Core Writing Team, Synthesis Report; Pachauri, Rajendra K.; Meyer, Leo; Pl.

Klaus, V.H., Kleinebecker, T., Prati, D., Gossner, M.M., Alt, F., Boch, S., Gockel, S., Hemp, A., Lange, M., Müller, J., Oelmann, Y., Pašalić, E., Renner, S.C., Socher, S.A., Türke, M., Weisser, W.W., Fischer, M., Hölzel, N., 2013. Does organic grassland farming benefit plant and arthropod diversity at the expense of yield and soil fertility? *Agric.*

601 Ecosyst. Environ. 177, 1–9. <https://doi.org/10.1016/j.agee.2013.05.019>

602 Koricheva, J., Mulder, C.P.H., Schmid, B., Joshi, J., Huss-Danell, K., 2000. Numerical  
603 responses of different trophic groups of invertebrates to manipulations of plant diversity  
604 in grasslands. *Oecologia* 125, 271–282. <https://doi.org/10.1007/s004420000450>

605 Kremen, C., Williams, N.M., Aizen, M.A., Gemmill-Herren, B., LeBuhn, G., Minckley, R.,  
606 Packer, L., Potts, S.G., Roulston, T., Steffan-Dewenter, I., Vázquez, D.P., Winfree, R.,  
607 Adams, L., Crone, E.E., Greenleaf, S.S., Keitt, T.H., Klein, A.M., Regetz, J., Ricketts,  
608 T.H., 2007. Pollination and other ecosystem services produced by mobile organisms: A  
609 conceptual framework for the effects of land-use change. *Ecol. Lett.* 10, 299–314.  
610 <https://doi.org/10.1111/j.1461-0248.2007.01018.x>

611 Leather, S.R., 2018. “Ecological Armageddon” - more evidence for the drastic decline in  
612 insect numbers. *Ann. Appl. Biol.* 172, 1–3. <https://doi.org/10.1111/aab.12410>

613 Lichtenberg, E.M., Kennedy, C.M., Kremen, C., Batáry, P., Berendse, F., Bommarco, R.,  
614 Bosque-Pérez, N.A., Carvalheiro, L.G., Snyder, W.E., Williams, N.M., Winfree, R.,  
615 Klatt, B.K., Åström, S., Benjamin, F., Brittain, C., Chaplin-Kramer, R., Clough, Y.,  
616 Danforth, B., Diekötter, T., Eigenbrode, S.D., Ekroos, J., Elle, E., Freitas, B.M., Fukuda,  
617 Y., Gaines-Day, H.R., Grab, H., Gratton, C., Holzschuh, A., Isaacs, R., Isaia, M., Jha, S.,  
618 Jonason, D., Jones, V.P., Klein, A.-M., Krauss, J., Letourneau, D.K., Macfadyen, S.,  
619 Mallinger, R.E., Martin, E.A., Martinez, E., Memmott, J., Morandin, L., Neame, L.,  
620 Otieno, M., Park, M.G., Pfiffner, L., Pocock, M.J.O., Ponce, C., Potts, S.G., Poveda, K.,  
621 Ramos, M., Rosenheim, J.A., Rundlöf, M., Sardiñas, H., Saunders, M.E., Schon, N.L.,  
622 Sciligo, A.R., Sidhu, C.S., Steffan-Dewenter, I., Tschardtke, T., Veselý, M., Weisser,  
623 W.W., Wilson, J.K., Crowder, D.W., 2017. A global synthesis of the effects of  
624 diversified farming systems on arthropod diversity within fields and across agricultural

625 landscapes. *Glob. Chang. Biol.* 23, 4946–4957. <https://doi.org/10.1111/gcb.13714>

626 Mantyka-Pringle, C.S., Visconti, P., Di Marco, M., Martin, T.G., Rondinini, C., Rhodes, J.R.,  
 627 2015. Climate change modifies risk of global biodiversity loss due to land-cover change.  
 628 *Biol. Conserv.* 187, 103–111. <https://doi.org/10.1016/j.biocon.2015.04.016>

629 Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Synthesis*.  
 630 Island Press, Washington DC. <https://doi.org/10.1196/annals.1439.003>

631 Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L.,  
 632 Bennett, D.J., Choimes, A., Collen, B., 2015. Global effects of land use on local  
 633 terrestrial biodiversity. *Nature* 520, 45–50.

634 Norberg, J., Swaney, D.P., Dushoff, J., Lin, J., Casagrandi, R., Levin, S.A., 2001. Phenotypic  
 635 diversity and ecosystem functioning in changing environments: A theoretical  
 636 framework. *Proc. Natl. Acad. Sci. U. S. A.* 98, 11376–11381.  
 637 <https://doi.org/10.1073/pnas.171315998>

638 Öckinger, E., Smith, H.G., 2006. Semi-natural grasslands as population sources for  
 639 pollinating insects in agricultural landscapes. *J. Appl. Ecol.* 44, 50–59.  
 640 <https://doi.org/10.1111/j.1365-2664.2006.01250.x>

641 Oliver, T.H., Morecroft, M.D., 2014. Interactions between climate change and land use  
 642 change on biodiversity: attribution problems, risks, and opportunities. *Wiley Interdiscip.*  
 643 *Rev. Clim. Chang.* 5, 317–335. <https://doi.org/10.1002/wcc.271>

644 Pilgrim, E.S., Macleod, C.J.A., Blackwell, M.S.A., Bol, R., Hogan, D. V., Chadwick, D.R.,  
 645 Cardenas, L., Misselbrook, T.H., Haygarth, P.M., Brazier, R.E., Hobbs, P., Hodgson, C.,  
 646 Jarvis, S., Dungait, J., Murray, P.J., Firbank, L.G., 2010. Interactions Among  
 647 Agricultural Production and Other Ecosystem Services Delivered from European

648 Temperate Grassland Systems. *Adv. Agron.* 109, 117–154.

649 <https://doi.org/10.1016/B978-0-12-385040-9.00004-9>

650 R Core Team, 2020. R: A Language and Environment for Statistical Computing.

651 Robinson, S.I., McLaughlin, Ó.B., Marteinsdóttir, B., O’Gorman, E.J., 2018. Soil

652 temperature effects on the structure and diversity of plant and invertebrate communities

653 in a natural warming experiment. *J. Anim. Ecol.* 87, 634–646.

654 <https://doi.org/10.1111/1365-2656.12798>

655 Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald,

656 E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney,

657 H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H.,

658 2000. Global biodiversity scenarios for the year 2100. *Science* 287, 1770–1774.

659 Schädler, M., Buscot, F., Klotz, S., Reitz, T., Durka, W., Bumberger, J., Merbach, I.,

660 Michalski, S.G., Kirsch, K., Remmler, P., Schulz, E., Auge, H., 2019. Investigating the

661 consequences of climate change under different land- use regimes: a novel experimental

662 infrastructure. *Ecosphere* 10, e02635. <https://doi.org/10.1002/ecs2.2635>

663 Scheffers, B.R., De Meester, L., Bridge, T.C.L., Hoffmann, A.A., Pandolfi, J.M., Corlett,

664 R.T., Butchart, S.H.M., Pearce-Kelly, P., Kovacs, K.M., Dudgeon, D., Pacifici, M.,

665 Rondinini, C., Foden, W.B., Martin, T.G., Mora, C., Bickford, D., Watson, J.E.M.,

666 2016. The broad footprint of climate change from genes to biomes to people. *Science*

667 354. <https://doi.org/10.1126/science.aaf7671>

668 Scherber, C., Eisenhauer, N., Weisser, W.W., Schmid, B., Voigt, W., Fischer, M., Schulze,

669 E.D., Roscher, C., Weigelt, A., Allan, E., Beler, H., Bonkowski, M., Buchmann, N.,

670 Buscot, F., Clement, L.W., Ebeling, A., Engels, C., Halle, S., Kertscher, I., Klein, A.M.,

671 Koller, R., König, S., Kowalski, E., Kummer, V., Kuu, A., Lange, M., Lauterbach, D.,

672 Middelhoff, C., Migunova, V.D., Milcu, A., Müller, R., Partsch, S., Petermann, J.S.,  
 673 Renker, C., Rottstock, T., Sabais, A., Scheu, S., Schumacher, J., Temperton, V.M.,  
 674 Tschamtk, T., 2010. Bottom-up effects of plant diversity on multitrophic interactions in  
 675 a biodiversity experiment. *Nature* 468, 553–556. <https://doi.org/10.1038/nature09492>

676 Schuldt, A., Ebeling, A., Kunz, M., Staab, M., Guimarães-Steinicke, C., Bachmann, D.,  
 677 Buchmann, N., Durka, W., Fichtner, A., Fornoff, F., Härdtle, W., Hertzog, L.R., Klein,  
 678 A.-M., Roscher, C., Schaller, J., von Oheimb, G., Weigelt, A., Weisser, W., Wirth, C.,  
 679 Zhang, J., Bruehlheide, H., Eisenhauer, N., 2019. Multiple plant diversity components  
 680 drive consumer communities across ecosystems. *Nat. Commun.* 10, 1–11.  
 681 <https://doi.org/10.1038/s41467-019-09448-8>

682 Seibold, S., Gossner, M.M., Simons, N.K., Blüthgen, N., Müller, J., Ambarlı, D., Ammer, C.,  
 683 Bauhus, J., Fischer, M., Habel, J.C., Linsenmair, K.E., Nauss, T., Penone, C., Prati, D.,  
 684 Schall, P., Schulze, E.-D., Vogt, J., Wöllauer, S., Weisser, W.W., 2019. Arthropod  
 685 decline in grasslands and forests is associated with landscape-level drivers. *Nature* 574,  
 686 671–674. <https://doi.org/10.1038/s41586-019-1684-3>

687 Simons, N.K., Gossner, M.M., Lewinsohn, T.M., Lange, M., Türke, M., Weisser, W.W.,  
 688 2015. Effects of land-use intensity on arthropod species abundance distributions in  
 689 grasslands. *J. Anim. Ecol.* 84, 143–154. <https://doi.org/10.1111/1365-2656.12278>

690 Smith, B., Wilson, J.B., 1996. A Consumer's Guide to Evenness Indices. *Oikos* 76, 70–82.  
 691 <https://doi.org/10.2307/3545749>

692 Soliveres, S., van der Plas, F., Manning, P., Prati, D., Gossner, M.M., Renner, S.C., Alt, F.,  
 693 Arndt, H., Baumgartner, V., Binkenstein, J., Birkhofer, K., Blaser, S., Blüthgen, N.,  
 694 Boch, S., Böhm, S., Börschig, C., Buscot, F., Diekötter, T., Heinze, J., Hölzel, N., Jung,  
 695 K., Klaus, V.H., Kleinebecker, T., Klemmer, S., Krauss, J., Lange, M., Morris, E.K.,

696 Müller, J., Oelmann, Y., Overmann, J., Pašalić, E., Rillig, M.C., Schaefer, H.M.,  
 697 Schloter, M., Schmitt, B., Schöning, I., Schrumpf, M., Sikorski, J., Socher, S.A., Solly,  
 698 E.F., Sonnemann, I., Sorkau, E., Steckel, J., Steffan-Dewenter, I., Stempfhuber, B.,  
 699 Tschapka, M., Türke, M., Venter, P.C., Weiner, C.N., Weisser, W.W., Werner, M.,  
 700 Westphal, C., Wilcke, W., Wolters, V., Wubet, T., Wurst, S., Fischer, M., Allan, E.,  
 701 2016. Biodiversity at multiple trophic levels is needed for ecosystem multifunctionality.  
 702 Nature 536, 456–459. <https://doi.org/10.1038/nature19092>

703 Soroye, P., Newbold, T., Kerr, J., 2020. Climate change contributes to widespread declines  
 704 among bumble bees across continents. Science 367, 685–688.  
 705 <https://doi.org/10.1126/science.aax8591>

706 Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M.C., Schwager, M., Jeltsch, F.,  
 707 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance  
 708 of keystone structures. J. Biogeogr. 31, 79–92. [https://doi.org/10.1046/j.0305-](https://doi.org/10.1046/j.0305-0270.2003.00994.x)  
 709 [0270.2003.00994.x](https://doi.org/10.1046/j.0305-0270.2003.00994.x)

710 Thomas, C.F., Marshall, E.J., 1999. Arthropod abundance and diversity in differently  
 711 vegetated margins of arable fields. Agric. Ecosyst. Environ. 72, 131–144.  
 712 [https://doi.org/10.1016/S0167-8809\(98\)00169-8](https://doi.org/10.1016/S0167-8809(98)00169-8)

713 Thompson, R.M., Beardall, J., Beringer, J., Grace, M., Sardina, P., 2013. Means and  
 714 extremes: Building variability into community-level climate change experiments. Ecol.  
 715 Lett. 16, 799–806. <https://doi.org/10.1111/ele.12095>

716 Tilman, D., El Haddi, A., 1992. Drought and biodiversity in Grasslands. Oecologia 89, 257–  
 717 264. <https://doi.org/10.1007/BF00317226>

718 van Klink, R., Bowler, D.E., Gongalsky, K.B., Swengel, A.B., Gentile, A., Chase, J.M.,  
 719 2020. Meta-analysis reveals declines in terrestrial but increases in freshwater insect

abundances. *Science* 368, 417–420. <https://doi.org/10.1126/SCIENCE.AAX9931>

van Klink, R., Menz, M.H.M., Baur, H., Dosch, O., Kühne, I., Lischer, L., Luka, H., Meyer, S., Szikora, T., Unternährer, D., Arlettaz, R., Humbert, J.Y., 2019. Larval and phenological traits predict insect community response to mowing regime manipulations. *Ecol. Appl.* 29, e01900. <https://doi.org/10.1002/eap.1900>

Vitousek, P.M., 1994. Beyond Global Warming: Ecology and Global Change. *Ecology* 75, 1861–1876. <https://doi.org/10.2307/1941591>

von Berg, K., Thies, C., Tschardt, T., Scheu, S., 2009. Cereal aphid control by generalist predators in presence of belowground alternative prey: Complementary predation as affected by prey density. *Pedobiologia (Jena)*. 53, 41–48. <https://doi.org/10.1016/j.pedobi.2009.03.001>

## Figure Captions

**Figure 1.** Aerial view of the experimental units of the GCEF showing two of the climate units in the foreground with their five land-use treatments. The arthropod communities within the experimental units of the GCEF (small boxes) are assembled through colonisation from the regional species pool (large box) which is influenced by current climatic conditions. (Picture modified from A. Künzelmann/UFZ; insect silhouettes from [www.all-silhouettes.com](http://www.all-silhouettes.com))

**Figure 2.** Abundance, species richness and evenness of the whole community across climate change and land-use intensity (LUI) treatments. Lines and asterisks indicate statistically significant differences between climate and land-use intensity treatments (\*\*\*  $p < 0.001$ , \*\*  $p$



<0.01. \*  $p < 0.05$ ), derived from estimated marginal contrasts (for interpretive aid only; see Table 1 for results from GLMMs). Colours correspond to land-use intensity treatments, with green representing low intensity managed grasslands and orange representing high intensity managed grasslands.

**Figure 3.** Abundance, species richness and evenness of predators, herbivores, omnivores and detritivores across climate change and land-use intensity (LUI) treatments. Lines and asterisks indicate statistically significant differences between climate and land-use intensity treatments (\*\*\*  $p < 0.001$ , \*\*  $p < 0.01$ . \*  $p < 0.05$ ), derived from estimated marginal contrasts (for interpretive aid only; see Table 1 for results from GLMMs). Lines and asterisks highlighted blue indicate a significant interaction between land-use intensity and climate change. Colours correspond to land-use intensity treatments, with green representing low intensity managed grasslands and orange representing high intensity managed grasslands.

**Figure 4.** Non-metric multidimensional scaling (NMDS) plots for the whole community (A) and of the four major trophic groups, predators (B), herbivores (C), detritivores (D), and omnivores (E) based on Bray-Curtis dissimilarity. Orange polygons surround communities from high intensity grasslands and green polygons surround communities from low intensity grasslands. Solid and dashed lines denote current and future climate simulations, respectively.

763 **Table 1.** Estimates, standard errors and p-values from the generalised linear mixed effects models with whole-community abundance, species  
764 richness, and evenness as response variables and climate treatment (“Current” vs. “Future”), land-use intensity (LUI) treatment (low intensity  
765 managed grassland vs. high intensity managed grassland), and their interaction as predictors. Significant p-values are highlighted in bold.

	<b>Abundance</b>		<b>Species Richness</b>		<b>Evenness</b>	
	Estimate ± Std. Error	p-value	Estimate ± Std. Error	p-value	Estimate ± Std. Error	p-value
Intercept	6.761 ± 0.081	<b>&lt;0.001</b>	4.246 ± 0.054	<b>&lt;0.001</b>	0.478 ± 0.015	<b>&lt;0.001</b>
Future Climate	-0.269 ± 0.115	<b>0.019</b>	-0.003 ± 0.076	0.970	0.093 ± 0.021	<b>&lt;0.001</b>
LUI High	-0.53 ± 0.094	<b>&lt;0.001</b>	-0.189 ± 0.08	<b>0.018</b>	0.011 ± 0.018	0.570
Future × LUI High	0.12 ± 0.134	0.370	-0.058 ± 0.113	0.611	-0.042 ± 0.026	0.137

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768

769 **Table 2.** Estimates, standard errors and p-values from the generalised linear mixed effects models with abundance, species richness, and  
770 evenness of the four trophic groups as response variables and climate treatment (“Current” vs. “Future”), land-use intensity (LUI) treatment (low  
771 intensity managed grassland vs. high intensity managed grassland), and their interaction as predictors. Significant p-values are highlighted in  
772 bold.

Model	Predators		Herbivores		Detritivores		Omnivores	
	Estimate ± Std. Error	p-value	Estimate ± Std. Error	p-value	Estimate ± Std. Error	p-value	Estimate ± Std. Error	p-value
<b>Abundance</b>								
Intercept	5.222 ± 0.098	<b>&lt;0.001</b>	5.595 ± 0.128	<b>&lt;0.001</b>	4.144 ± 0.149	<b>&lt;0.001</b>	4.535 ± 0.189	<b>&lt;0.001</b>
Future Climate	-0.209 ± 0.139	0.134	-0.234 ± 0.181	0.196	-0.798 ± 0.220	<b>&lt;0.001</b>	-0.113 ± 0.267	0.671
LUI High	-0.602 ± 0.054	<b>0.000</b>	-0.154 ± 0.151	0.309	-0.814 ± 0.099	<b>&lt;0.001</b>	-0.155 ± 0.268	0.562
Future Climate × LUI High	0.206 ± 0.079	<b>0.009</b>	-0.004 ± 0.214	0.986	0.388 ± 0.164	<b>0.018</b>	0.17 ± 0.379	0.654
<b>Species richness</b>								
Intercept	2.721 ± 0.115	<b>&lt;0.001</b>	3.714 ± 0.070	<b>&lt;0.001</b>	2.104 ± 0.156	<b>&lt;0.001</b>	1.686 ± 0.192	<b>&lt;0.001</b>
Future Climate	0.064 ± 0.160	0.690	0.019 ± 0.098	0.844	-0.248 ± 0.236	0.293	-0.038 ± 0.275	0.891
LUI High	-0.141 ± 0.168	0.402	-0.293 ± 0.107	<b>0.006</b>	0.000 ± 0.221	1.000	0.071 ± 0.267	0.789
Future Climate × LUI High	0.09 ± 0.232	0.696	-0.137 ± 0.153	0.372	0.000 ± 0.334	1.000	-0.034 ± 0.383	0.930
<b>Evenness</b>								
Intercept	-0.17 ± 0.029	<b>&lt;0.001</b>	-0.351 ± 0.023	<b>&lt;0.001</b>	-0.335 ± 0.042	<b>&lt;0.001</b>	-0.201 ± 0.037	<b>&lt;0.001</b>
Future Climate	-0.007 ± 0.041	0.859	0.067 ± 0.032	0.052	0.148 ± 0.060	<b>0.026</b>	0.066 ± 0.053	0.234
LUI High	-0.04 ± 0.029	0.208	0.002 ± 0.028	0.940	0.16 ± 0.060	<b>0.017</b>	-0.098 ± 0.046	0.064
Future Climate × LUI High	-0.013 ± 0.041	0.755	-0.011 ± 0.039	0.787	-0.09 ± 0.085	0.305	-0.018 ± 0.064	0.788

**Table 3.** Results ( $R^2$  and p-values) from permutational analysis of variance models testing for the effects of climate treatment and land-use intensity (LUI) under different climate scenarios on Bray-Curtis dissimilarity of whole arthropod communities and individual trophic groups. Significant p-values are highlighted in bold.

	Community $R^2$ (p-value)	Predators $R^2$ (p-value)	Herbivores $R^2$ (p-value)	Detritivores $R^2$ (p-value)	Omnivores $R^2$ (p-value)
Climate	0.08 ( <b>0.004</b> )	0.08 ( <b>0.004</b> )	0.06 ( <b>0.004</b> )	0.12 ( <b>0.020</b> )	0.02 (0.208)
LUI	0.42 ( <b>0.002</b> )	0.13 ( <b>0.002</b> )	0.52 ( <b>0.002</b> )	0.17 ( <b>0.002</b> )	0.17 (0.079)
Climate $\times$ LUI	0.04 (0.377)	0.05 (0.252)	0.05 (0.294)	0.037 (0.632)	0.004 (0.979)

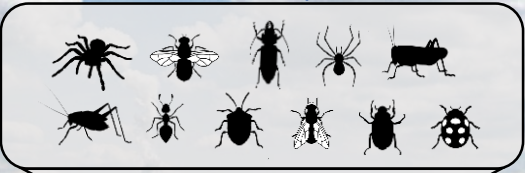
- We experimentally test for combined land-use and climate change impacts on arthropod communities
- Land-use and climate change reduced total abundance and diversity, but increased arthropod evenness
- Climate change interactively reduced land-use impacts on predator and detritivore abundance
- Land-use intensity had more widespread impacts on arthropods across trophic groups than climate change
- Both drivers caused shifts in species composition, within and across trophic groups

Figure 1

[Click here to access/download;Figure;Figure 1.pdf](#)



Species pool



Species assemblage

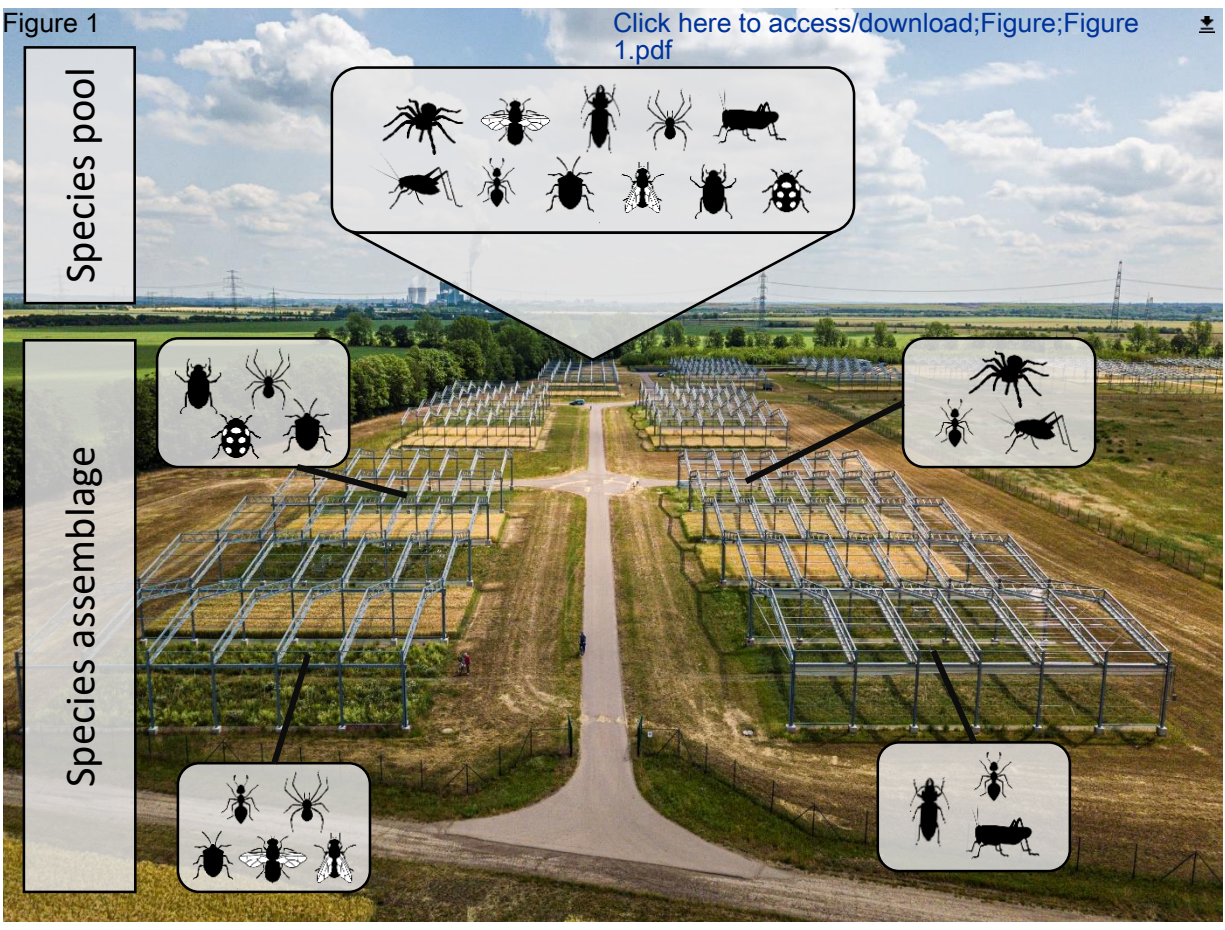


Figure 2

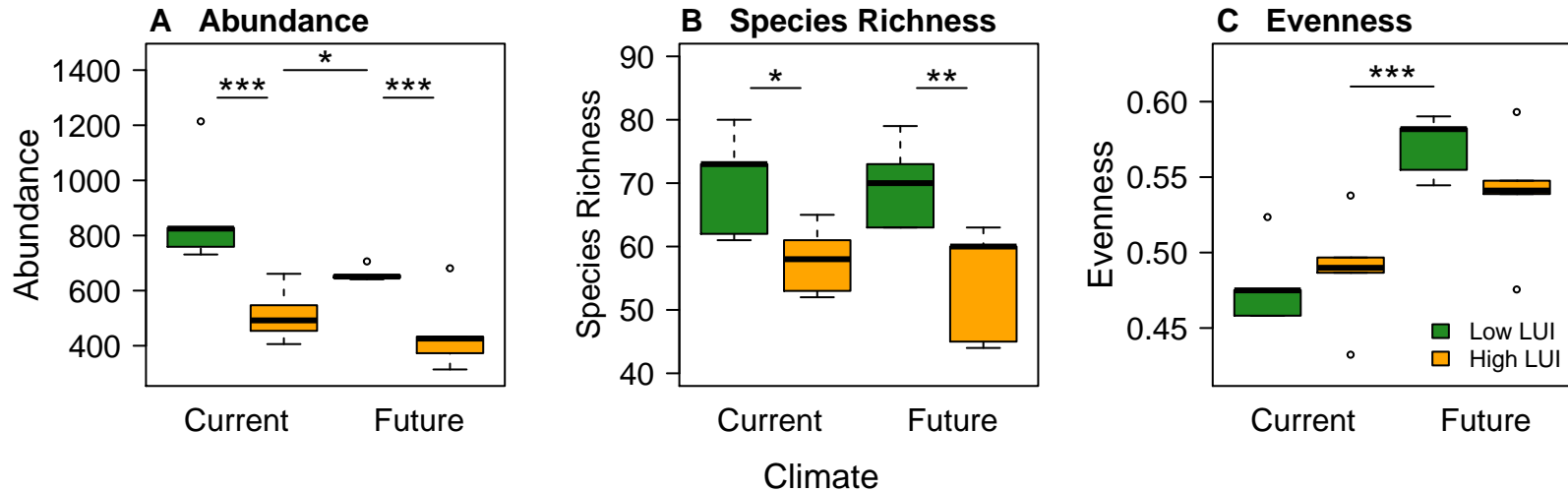
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Figure 3

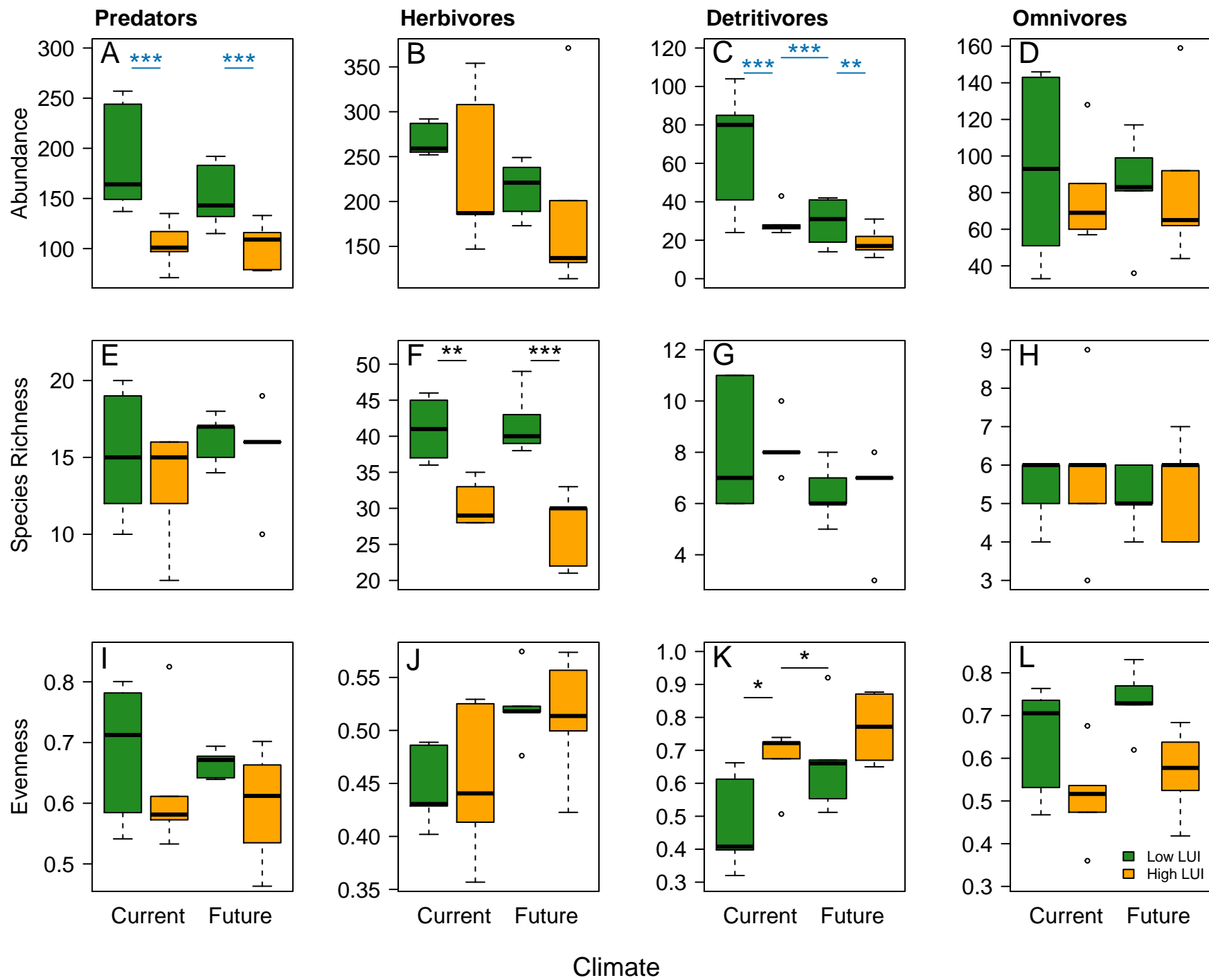
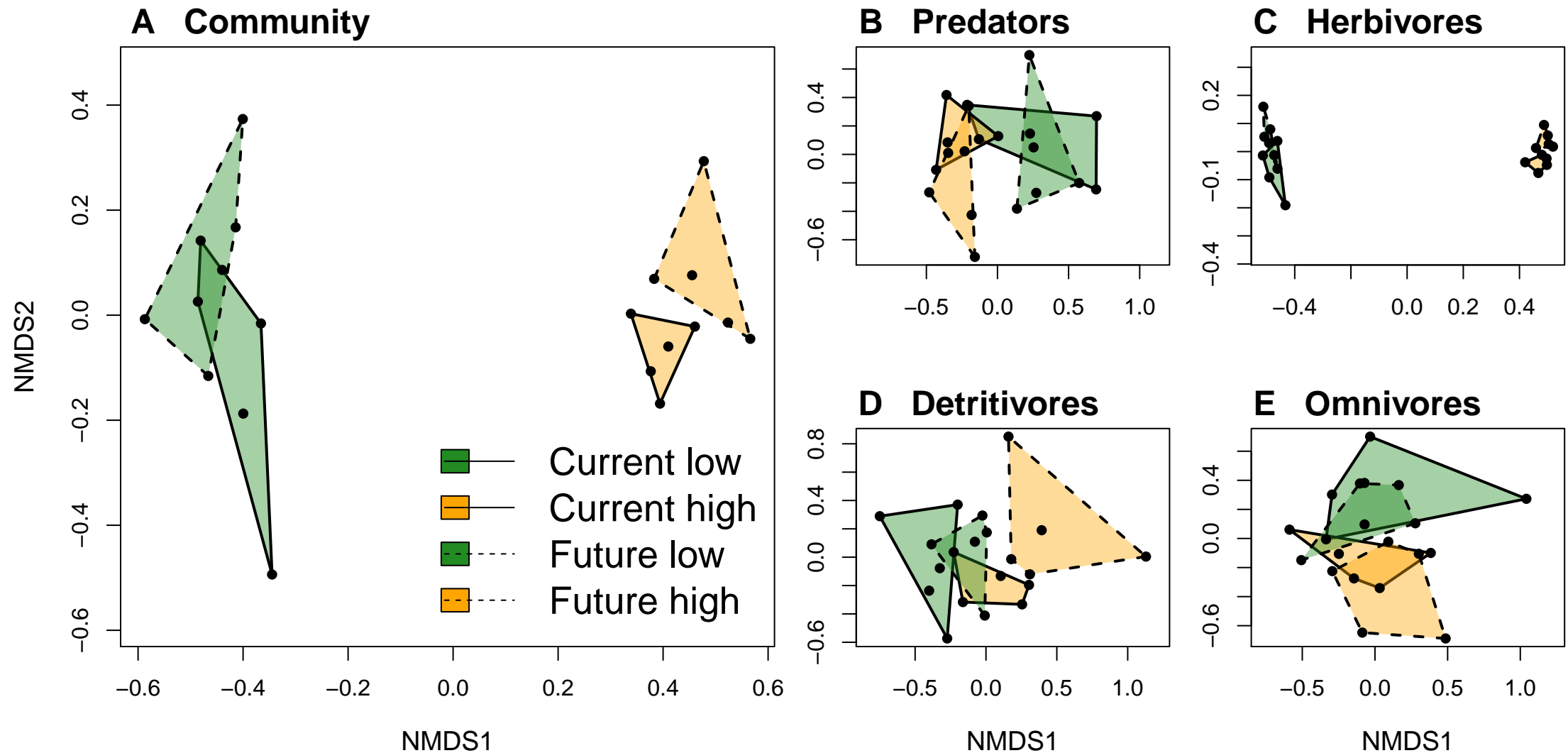




Figure 4



1 **Supporting Information**

2 Esra H. Sohlström, Ulrich Brose, Roel van Klink, Björn C. Rall, Benjamin Rosenbaum,  
3 Martin Schädler & Andrew D. Barnes. *Future climate and land-use intensification modify*  
4 *arthropod community composition*

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**Table S1.** Total abundances of each major taxonomic group collected within and across all treatments applied in our study: low and high land-use intensity (LUI) under both current and future climatic conditions. Taxa that were further identified by specialist taxonomists are indicated in bold.

Taxon	Current climate		Future climate		Total
	Low LUI	High LUI	Low LUI	High LUI	
<b>Araneae</b>	<b>307</b>	<b>243</b>	<b>260</b>	<b>227</b>	<b>1037</b>
<b>Auchenorrhyncha</b>	<b>772</b>	<b>669</b>	<b>547</b>	<b>652</b>	<b>2640</b>
Chilopoda	4	2	2	0	8
<b>Coleoptera</b>	<b>736</b>	<b>412</b>	<b>589</b>	<b>400</b>	<b>2137</b>
Dermaptera	4	24	4	3	35
Diplopoda	4	0	0	0	4
<b>Diptera</b>	<b>384</b>	<b>456</b>	<b>290</b>	<b>299</b>	<b>1429</b>
Heteroptera	124	53	51	76	304
<b>Hymenoptera</b>	<b>977</b>	<b>498</b>	<b>726</b>	<b>439</b>	<b>2640</b>
Isopoda	19	9	10	15	53
Lepidoptera	10	4	11	1	26
Opiliones	3	1	0	0	4
Orthoptera	2	5	6	4	17
Sternorrhyncha	581	89	250	64	984
Thysanoptera	600	193	638	146	1577

14 **Table S2.** Abundance (Abu.) and species richness (SR) of predators, herbivores, omnivores and  
15 detritivores across the five major taxonomic groups used in our analyses: Araneae, Auchenorrhyncha,  
16 Coleoptera, Diptera, and Hymenoptera.

<b>Araneae</b>								
Family	Predator		Herbivore		Omnivore		Detritivore	
	Abu.	SR	Abu.	SR	Abu.	SR	Abu.	SR
Araneidae	4	2	-	-	-	-	-	-
Dictynidae	1	1	-	-	-	-	-	-
Gnaphosidae	16	NA	-	-	-	-	-	-
Hahniidae	1	1	-	-	-	-	-	-
Linyphiidae	17	16	-	-	-	-	-	-
Lycosidae	5	2	-	-	-	-	-	-
Philodromidae	1	NA	-	-	-	-	-	-
Phrurolithidae	1	1	-	-	-	-	-	-
Pisauridae	1	1	-	-	-	-	-	-
Salticidae	1	1	-	-	-	-	-	-
Tetragnathidae	2	1	-	-	-	-	-	-
Theridiidae	3	1	-	-	-	-	-	-
Thomisidae	4	3	-	-	-	-	-	-

<b>Auchenorrhyncha</b>								
Family	Predator		Herbivore		Omnivore		Detritivore	
	Abu.	SR	Abu.	SR	Abu.	SR	Abu.	SR
Aphrophoridae	-	-	1	1	-	-	-	-
Cicadellidae	-	-	1994	18	-	-	-	-
Delphacidae	-	-	596	7	-	-	-	-

<b>Coleoptera</b>								
Family	Predator		Herbivore		Omnivore		Detritivore	
	Abu.	SR	Abu.	SR	Abu.	SR	Abu.	SR
Anobiidae	-	-	1	1	-	-	-	-
Anthicidae	-	-	1	1	-	-	-	-
Apionidae	-	-	189	15	-	-	-	-
Carabidae	133	12	-	-	9	9	-	-
Chrysomelidae	-	-	249	18	-	-	-	-
Coccinellidae	14	4	-	-	-	-	3	3
Corylophidae	-	-	-	-	-	-	50	1
Cryptophagidae	-	-	-	-	-	-	96	6
Curculionidae	-	-	361	25	-	-	-	-
Dermestidae	-	-	-	-	-	-	1	1
Elateridae	1	1	8	1	-	-	-	-
Hydrophilidae	-	-	-	-	1	1	-	-
Latridiidae	-	-	-	-	-	-	216	5
Leiodidae	-	-	-	-	-	-	2	2
Malachiidae	-	-	1	1	-	-	-	-
Mordellidae	-	-	9	4	-	-	-	-
Nitidulidae	-	-	137	4	-	-	-	-
Oedemeridae	-	-	1	1	-	-	-	-
Phalacridae	-	-	15	4	-	-	-	-

Scarabaeidae	-	-	-	-	-	-	1	1
Staphylinidae	235	19	-	-	404	11	-	-
Throscidae	-	-	1	1	-	-	-	-

## Diptera

Family	Predator		Herbivore		Omnivore		Detritivore	
	Abu.	SR	Abu.	SR	Abu.	SR	Abu.	SR
Agromyzidae	-	-	12	6	-	-	-	-
Anthomyiidae	-	-	8	4	-	-	-	-
Anthomyzidae	-	-	80	2	-	-	-	-
Biobionidae	-	-	1	1	-	-	-	-
Calliphoridae	-	-	2	1	-	-	-	-
Camillidae	-	-	4	1	-	-	-	-
Cecidomyiidae	-	-	240	NA	-	-	-	-
Ceratopogonidae	-	-	2	NA	-	-	-	-
Chaoboridae	-	-	1	1	-	-	-	-
Chironomidae	-	-	97	NA	-	-	-	-
Chloropidae	-	-	390	21	-	-	-	-
Conopidae	-	-	1	1	-	-	-	-
Dolichopodidae	7	1	8	2	-	-	-	-
Drosophilidae	-	-	-	-	9	1	-	-
Empididae	-	-	-	-	11	1	-	-
Ephydriidae	-	-	46	5	-	-	-	-
Hybotidae	6	4	-	-	1	-	-	-
Keroplatidae	-	-	21	2	-	-	-	-
Limoniidae	-	-	1	1	-	-	-	-
Lonchopteridae	-	-	26	2	-	-	-	-
Opomyzidae	-	-	12	3	-	-	-	-
Phoridae	2	1	-	-	39	3	-	-
Rhinophoridae	1	1	-	-	-	-	-	-
Sarcophagidae	-	-	2	1	-	-	-	-
Sciaridae	-	-	-	-	-	-	135	NA
Sepsidae	-	-	5	2	-	-	-	-
Sphaeroceridae	-	-	-	-	-	-	221	6
Syrphidae	-	-	8	3	-	-	-	-
Tachinidae	-	-	1	1	-	-	-	-
Tephritidae	-	-	2	2	-	-	-	-

## Hymenoptera

Family	Predator		Herbivore		Omnivore		Detritivore	
	Abu.	SR	Abu.	SR	Abu.	SR	Abu.	SR
Aphelinidae	18	NA	-	-	-	-	-	-
Apidae	-	-	3	2	-	-	-	-
Bethylidae	2	NA	-	-	-	-	-	-
Braconidae	62	NA	-	-	39	NA	-	-
Ceraphronidae	277	NA	-	-	-	-	-	-
Cynipidae	-	-	8	NA	-	-	-	-
Diapriidae	109	NA	-	-	-	-	-	-
Dryinidae	1	NA	-	-	-	-	-	-
Encyrtidae	28	NA	-	-	-	-	-	-
Eucoliidae	4	NA	-	-	-	-	-	-
Eulophidae	172	NA	-	-	-	-	-	-

Eupelmidae	2	NA	-	-	-	-	-	-
Eurytomidae	1	NA	-	-	10	NA	-	-
Formicidae	-	-	-	-	1181	NA	-	-
Ichneumonidae	51	NA	-	-	-	-	-	-
Megaspilidae	7	NA	-	-	-	-	-	-
Myrmaridae	211	NA	-	-	-	-	-	-
Ormyridae	1	NA	-	-	-	-	-	-
Platygastridae	43	NA	-	-	-	-	-	-
Pteromalidae	155	NA	-	-	-	-	-	-
Scelionidae	164	NA	-	-	-	-	-	-
Sphecidae	2	2	-	-	-	-	-	-
Tenthredinidae	-	-	6	NA	-	-	-	-
Torymidae	4	NA	-	-	-	-	-	-
Trichogrammatidae	4	NA	-	-	-	-	-	-

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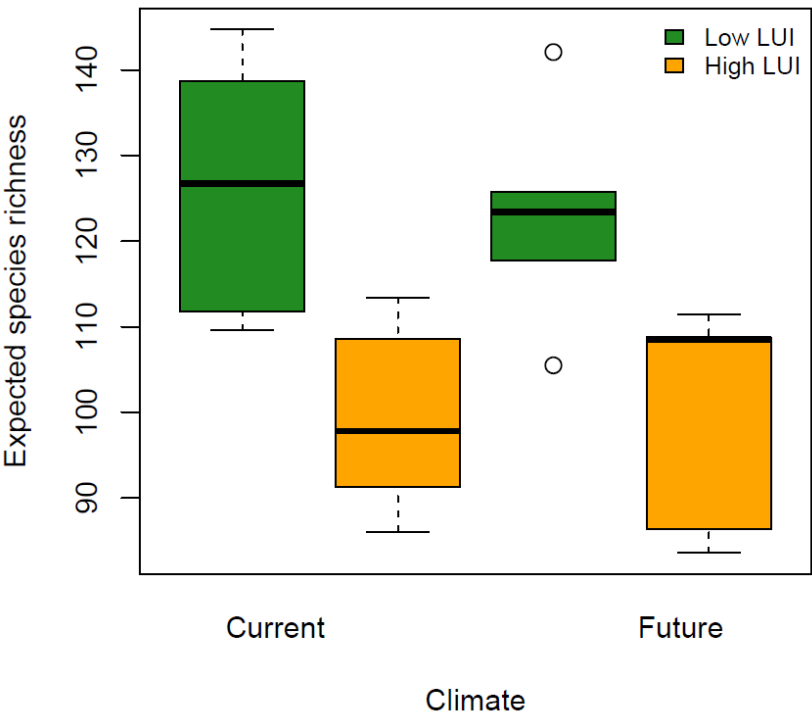
**Table S3.** Similarity percentage analysis (SIMPER) based on comparisons for A) low versus high land-use intensity and for B) current versus future climate treatments. Only taxa from a cumulative 50% contribution cut-off are shown.

**A) Land-use intensity**

Species	Mean abundances		Contribution to mean dissimilarity	Cumulative contribution to dissimilarity
	LUI	LUI		
	Low	High		
<i>Javesella pellucida</i>	13.70	36.60	0.05	0.07
<i>Aphrodes makarovi</i>	23.70	0.20	0.04	0.12
<i>Megophthalmus scanicus</i>	13.60	0.10	0.03	0.16
<i>Zyginidia scutellaris</i>	0.30	13.10	0.02	0.19
<i>Sitona lineatus</i>	13.50	0.90	0.02	0.22
<i>Atheta (Mocyta) cf. amplicollis</i>	3.80	16.10	0.02	0.26
<i>Pteremis fenestralis</i>	14.60	5.00	0.02	0.29
<i>Incertella albipalpis</i>	0.30	12.10	0.02	0.32
<i>Euscelis incisus</i>	12.70	1.10	0.02	0.34
<i>Eupteryx notata</i>	11.30	0.00	0.02	0.37
<i>Anaceratagallia ribauti</i>	10.60	0.00	0.02	0.40
<i>Conioscinella cf. frontella</i>	0.20	10.60	0.02	0.43
<i>Tenuiphantes tenuis</i>	9.40	12.20	0.01	0.44
<i>Microlestes maurus</i>	4.00	7.80	0.01	0.46
<i>Anoscopus serratulae</i>	5.70	11.20	0.01	0.48
<i>Protapion trifolii</i>	7.20	0.00	0.01	0.50

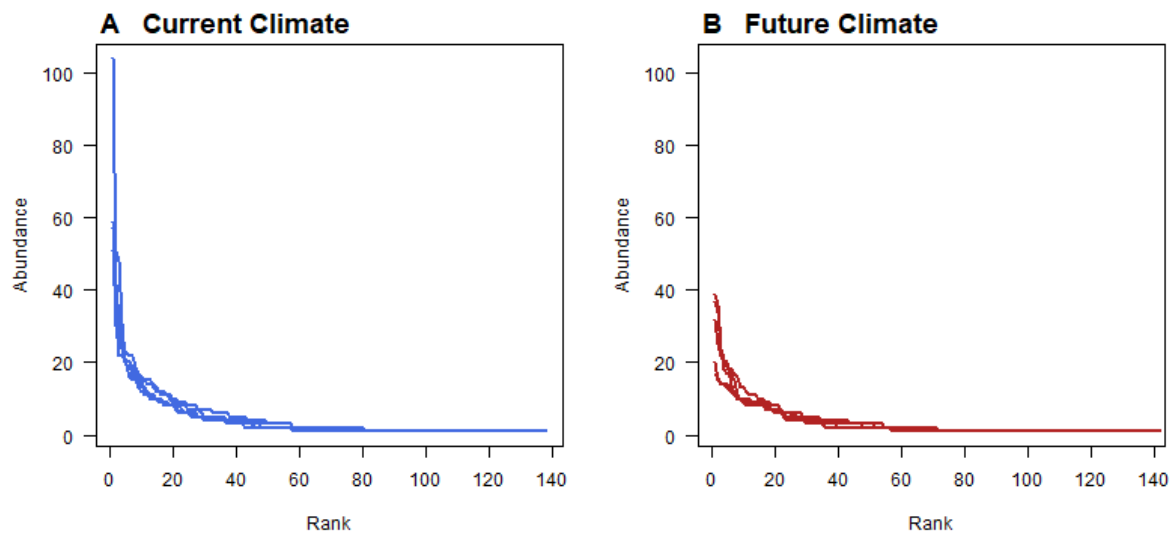
**B) Climate change**

Species	Mean abundances		Contribution to mean dissimilarity	Cumulative contribution to dissimilarity
	Current climate	Future climate		
<i>Javesella pellucida</i>	38.70	11.60	0.05	0.09
<i>Aphrodes makarovi</i>	13.70	10.20	0.03	0.13
<i>Pteremis fenestralis</i>	14.50	5.10	0.02	0.17
<i>Atheta (Mocyta) cf. amplicollis</i>	10.20	9.70	0.02	0.20
<i>Eupteryx notata</i>	9.00	2.30	0.02	0.22
<i>Megophthalmus scanicus</i>	5.80	7.90	0.02	0.25
<i>Zyginidia scutellaris</i>	7.60	5.80	0.02	0.28
<i>Incertella albipalpis</i>	4.40	8.00	0.02	0.30
<i>Euscelis incisus</i>	8.00	5.80	0.01	0.32
<i>Microlestes maurus</i>	2.60	9.20	0.01	0.35
<i>Sitona lineatus</i>	7.90	6.50	0.01	0.37
<i>Tenuiphantes tenuis</i>	10.30	11.30	0.01	0.39
<i>Stiphrosoma sabulosum</i>	7.50	0.40	0.01	0.41
<i>Meligethes aeneus</i>	6.90	2.30	0.01	0.43
<i>Anaceratagallia ribauti</i>	6.40	4.20	0.01	0.45
<i>Anoscopus serratulae</i>	8.30	8.60	0.01	0.47
<i>Conioscinella cf. frontella</i>	5.40	5.40	0.01	0.49
<i>Tachyporus hypnorum</i>	8.70	4.10	0.01	0.51



27 **Figure S1.** Expected species richness calculated using the jackknife 2-estimator across  
28 climate change and land-use (LUI) intensity treatments. Colours correspond to land-use  
29 intensity treatments, with green representing low intensity managed grasslands and orange  
30 representing high intensity managed grasslands.





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33 **Figure S2.** Rank abundance curves of the entire community within each replicate plot  
34 (denoted by individual curves) under (A) current and (B) future climate treatments.

**Declaration of interests**

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: